Sprinkling infiltration as an artificial groundwater recharge method - Long-term effects on boreal forest soil, tree growth and understory vegetation

Kaarakka, Lilli

2019-09-15

Kaarakka , L , Smolander , A , Lindroos , A-J , Nojd , P , Korpela , L , Nieminen , T M & Helmisaari , H-S 2019 , ' Sprinkling infiltration as an artificial groundwater recharge method - Long-term effects on boreal forest soil, tree growth and understory vegetation ' , Forest Ecology and Management , vol. 448 , pp. 240-248 . https://doi.org/10.1016/j.foreco.2019.06.015

http://hdl.handle.net/10138/305934
https://doi.org/10.1016/j.foreco.2019.06.015

Forest Ecology and Management 448 (2019) 240–248

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Sprinkling infiltration as an artificial groundwater recharge method – Long-term effects on boreal forest soil, tree growth and understory vegetation

Lilli Kaarakka[a,*], Aino Smolander[b], Antti-Jussi Lindroos[b], Pekka Nöjd[b], Leila Korpela[b], Tiina M. Nieminen[b], Heljä-Sisko Helmisaari[a]

[a] Department of Forest Sciences, University of Helsinki, P.O. Box 27, 00014 Helsinki, Finland
[b] Natural Resources Institute Finland (Luke), Latokartanonkaari 9, 00790 Helsinki, Finland

ARTICLE INFO

Keywords:
Artificial groundwater recharge
Forest soil
Sprinkling infiltration
Forest vegetation
Soil disturbance
Soil recovery

ABSTRACT

The artificial recharge of groundwater by infiltrating surface water through forest soil has been introduced as a groundwater producing practice in Finland. As a result, the forest soil, as well as the whole ecosystem, is subjected to extremely high inputs of carbon and nutrient rich lake water.

The effects of sprinkling infiltration on forest soil, tree growth and understory vegetation and their respective recovery were studied on a forested esker in central Finland. The Scots pine-dominated experimental plots were sprinkled with lake water in 1998–2001 and sampled after a 12–15-year recovery period. Soil pH and base cation concentration, as well as the rate of net N mineralization were significantly higher at the plots that had been infiltrated. The concentrations of base cations calcium and magnesium were thousands of times higher in the infiltrated soil than in the untreated soil. In addition, sprinkling infiltration had favored early-successional herbs, grasses and forbs and negatively affected late successional, slow-growing mosses and lichens. Sprinkling infiltration had significantly increased tree radial growth.

Sprinkling infiltration is an environment altering soil treatment method which, based on the findings of this study, can have long-term effects on tree growth, soil processes and understory vegetation.

1. Introduction

Global water consumption has significantly increased in the last century, and water scarcity and overexploitation of freshwater resources are major concerns in the international discourse (Kummu et al., 2016). Finland, like the other Nordic countries, has extensive groundwater reserves, and approximately 63% of the water distributed by public waterworks is groundwater (GTK, 2016). The majority of groundwater recharge occurs on eskers and moraine formations as rain, snowmelt and surface water infiltrate through the soil profile to the vadose zone. Groundwater reserves are, however, often scattered and not large enough to accommodate larger cities. Thus, many cities in Finland are using or planning to produce artificial groundwater by infiltration of surface water originating from lakes and rivers (Kätkö et al., 2006). Sprinkling infiltration differs from other infiltration methods, in that surface water is sprinkled directly onto the forest floor via a network of pipes without removing the vegetation and soil, as in basin recharge. Typically, Finnish surface waters are rich in organic matter as a result of run-off from terrestrial ecosystems, from both mineral soils and peatlands. However, groundwater which is formed from natural precipitation filtering through forest soil, is nearly free of organic carbon (C) and is therefore as considered ready-to-use household water (Lindroos et al., 2002)

During sprinkling infiltration, forest soil is subjected to extremely large inputs of water, in which relatively large quantities of nutrients such as calcium, magnesium and nitrogen (Ca^{2+}, Mg^{2+}, N) are added onto the forest floor and into the soil. Therefore, one could expect that sprinkling infiltration alters the soil pH, nutrient and C dynamics, and species composition of the forest floor, as well as the growth of the forest stand. Previous studies in Finland have reported significant increases in soil pH and concentrations of base cations Ca^{2+} and Mg^{2+} thus indicating that sprinkling infiltration leads to the neutralization of the forest soil (Paavolainen et al., 2000a; Lindroos et al., 2001). This shift in soil acidity and nutrient concentrations in turn acts as a driver for acid-sensitive nitrogen (N) transformations such as nitrification (Paavolainen et al., 2000a). Finnish forest soils are typically acidic (pH 4–5) and availability of N is limiting tree growth, particularly on nutrient-poor, sandy soils (Paavolainen, 1999; Korhonen et al., 2013; Högberg et al., 2017). Plant-available N is added to the soil slowly through depolymerization of N-containing compounds in decomposing

* Corresponding author.
E-mail address: lilli.kaarakka@helsinki.fi (L. Kaarakka).

https://doi.org/10.1016/j.foreco.2019.06.015
Received 28 August 2018; Received in revised form 4 June 2019; Accepted 8 June 2019
Available online 17 June 2019

litter and organic matter and finally, by N mineralization. Net nitrification is usually negligible in undisturbed forest soils due to low availability of ammonium and low pH (Paavolainen and Smolander, 1998; Paavolainen et al., 2000a; Sponseller et al., 2016).

Vegetation responses to environmental change depend on the nature and intensity of the disturbance and on the environmental requirements of the species in question. Little is known on the effects of sprinkling infiltration on forest understory vegetation. The effects of atmospheric N deposition (Dirnböck et al., 2014) and forest management, in particular logging (Palviainen et al., 2005; Tarvainen et al., 2015; Tonteri et al., 2016) on plant community composition and abundance have been studied, however. Previous studies have emphasized the high site-dependency of the effects of a disturbance on plant community composition (Tonteri et al., 1990; Manninen et al., 2009; Tarvainen et al., 2015). Sprinkling infiltration can be considered a form of forest fertilization as nutrients are added onto the forest floor and soil in the sprinkled lake water.

This study is a part of a research project initiated by the former Finnish Forest Research Institute (since 2015 Natural Resources Institute Finland, Luke) at the end of the 1990s, which studied the effects of lake water infiltration on soil properties and processes, soil percolate water quality, tree growth and understory vegetation. The experiment was designed as a part of a sprinkling infiltration water plant, operated by the local waterworks in Vuontee, Central Finland. The initial findings of the experiments conducted between 1998 and 2003 in the experimental stand have been reported in peer-reviewed journals (Nöjd et al., 2009), and scientific and technical reports (Helmisaari et al., 2003; Derome et al., 2004, 2006; Helmisaari et al., 2006). In this paper, we present results that have not been published before. This study is a follow-up study, which builds on data collected at the start of the infiltration treatment, immediately after it and 12–14 years after the treatment was terminated.

The overall aim of this study was to determine the effects of chemical load on forest soil and understory plant community and tree growth and to evaluate the rate and direction of the recovery of the ecosystem from this disturbance. Sprinkling infiltration adds nutrient-rich, high-pH lake water to the soil surface which in quantity exceeds the annual precipitation by the thousands. Therefore, we hypothesize that soil recovery occurs relatively slowly over time as hydrogen releasing compounds in natural precipitation, throughfall and organic leachate from the surface of the soil acidify the soil layers. In addition, we hypothesize that sprinkling infiltration causes changes in the understory vegetation composition, potentially shifting the understory community to reflect a more mesic stand.

2. Material and methods

2.1. The experiment

The sprinkling infiltration study site is located in Vuontee (62°20′8″N, 26°2′5″E), Central Finland, where the Finnish Forest Research Institute together with the local waterworks established experimental plots on an forested esker that was going to be used for groundwater recharge. The infiltrated stand is a mature (125–130 yrs) sub-xeric Scots pine-dominated, *Vaccinium vitis-idaea* (VT) site *sensu* (Cajander, 1949). The stand had standing stock of 770 stems/ha and 320 m^3/ha, had not been thinned in several decades. The stand had regenerated in the end of the 1890s through natural regeneration (Nöjd et al., 2009).

The soil consists of relatively coarse sandy deposits and the soil type was identified as humic podzol. In a VT-type stand, plant species diversity is low and limited to few pre-dominant species (mosses such as *Pleurozium schreberi,* dwarf shrubs such *Vaccinium myrtillus* and *Calluna vulgaris*, and lichens).

The experimental stand was previously studied in 1998–2003, and in 2012–2015, as part of this current follow-up study. Water from a nearby lake was sprinkled directly onto the forest floor via a network of pipes. During the infiltration treatment, the amount of infiltrated water was 600 m^3 m^{-2} annually (ca. 600 000 – 1 000 000 mm), which is a 1000 times higher than annual precipitation (643 mm yr^{-1}) at the stand (Nöjd et al., 2009). Four experimental plots (30 × 30 m) were established in 1998; two of which were infiltrated during 20.9.1999–19.12.2001 (Helmisaari et al., 2003; Derome et al., 2004, 2006; Nöjd et al., 2009) and two remained as untreated controls until 2002. In order to ensure the comparability of the results, similar sampling methods were used in this current study (i.e. sampling in 2012–2015) as described by Derome et al. (2004) and Nöjd et al. (2009).

2.2. Soil sampling and laboratory analyses

In 2013, 20 soil samples (10 per plot) were collected from each experimental plot. The samples were collected from the organic layer and from the mineral soil (0–5, 5–10, 10–20, 20–40 cm depth) from the previously infiltrated plots and from the uninfiltrated control plots (n = 2). Samples were combined to form two composite samples per soil layer for each plot and dried at 50 °C. The organic layer samples were milled through a 1 mm and mineral soil samples through a 2 mm sieve.

Total C and N concentrations were determined directly from the homogenized soil samples with a VarioMax-analyzer. $pH_{(H2O)}$ was measured directly from the samples with a glass electrode (volume 3/5-ratio). Exchangeable cations were extracted with a 0.2 M barium chloride solution with a ratio of 2 g/organic matter and 10 g/mineral soil sample per 100 ml of $BaCl_2$. The samples were left to stand overnight and then shaken for two hours and filtered using Whatman-filter paper (S&S 5893). The concentrations of exchangeable cations Al^{3+}, Ca^{2+}, K^+ and Mg^{2+} were determined by ICP-OES at University of Helsinki. Exchangeable acidity (H^+) was determined by titrating an aliquot with 0.01 M NaOH solution to pH 7.

Separate soil samples for nitrogen transformation studies were collected in 2014 with a soil corer (diameter 58 mm); 16 samples were collected from the organic layer and 16 from the uppermost part of the mineral soil (0–10 cm) from each experimental plot (n = 2). The litter layer was not included in the sampling. The samples were stored in +5 °C and analyzed directly after transportation to the laboratory. For homogenization, fresh soil samples were sieved (with 2.8 mm mesh size).

Nitrogen transformations were studied in aerobic incubation experiments in the laboratory as described earlier (Priha and Smolander, 1997; Paavolainen et al., 2000a, 2000b). Two replicate soil samples (each 20 ml; 3–4 g of organic material and 17–19 g of mineral soil) were incubated for 40 days at a constant temperature (14 °C) and adjusted for 60% water-holding capacity (WHC) in 120 ml glass containers. WHC was measured by soaking the samples in water for 2 h and then draining them for 2 h (Priha and Smolander, 1997).

Samples were extracted with 40 ml 1 M KCl (shaken for 2 h, 200 rpm) and NH_4-N and NO_3-N concentrations were measured with a flow injection analyzer. The same extraction was done for the initial non-incubated soil samples. To calculate net N mineralization and net nitrification, initial NH_4 + NO_3-N or NO_3-N concentrations were subtracted from the final (post-incubation) concentrations, respectively. To describe the quality of organic matter, nitrogen results were expressed on organic matter basis (per kg of SOM).

2.3. Vegetation surveys and tree growth measurements

In order to understand the development and changes in the understory plant community during the infiltration and after it, understory vegetation was surveyed at the infiltrated and control plots. As a part of the previous study, vegetation surveys at the experimental site were completed before the start of infiltration in 1998; during the infiltration in 2000; directly after the termination in 2001; and a year and two years after the termination of the infiltration, in 2002 and 2003, respectively (Derome et al., 2004).

In 2015, understory plant species abundancies were visually estimated as proportional coverages using a 1 m × 1 m quadrat-form-frame, which was divided into 10 cm × 10 cm areas by strings, following the description of Derome et al. (2004). Vegetation composition was analyzed for nine quadrats at each experimental plot; three quadrats were evenly distributed on three lines across the plot. Plant species of both field layer and bottom layer (lichens and mosses) were identified and their percentage cover (0–100%) was visually estimated to a precision of 0.5%. In addition, the proportion of unvegetated surface of forest floor, litter (including dead wood), stumps, exposed mineral soil and rocks, was estimated.

The scientific names for vascular plants follow the nomenclature presented by Hämet-Ahti et al. (1998), for bryophytes that of Ulvinen et al. (2002) and for lichens Stenroos et al. (2011). In the data analysis, species were organized into plant functional groups. Tree seedlings and shrubs under 0.5 m in height were included in the same group with dwarf shrubs. Field layer vegetation (grasses, herbs and forbs) formed one group, and bottom layer vegetation, consisting of mosses, liverworts and lichens, was divided into two groups: mosses and lichens. The percent covers of all identified species within a functional group were summed up to give cover values for the functional groups.

In order to study tree radial growth, a total of 15 pines were cored in August 2013. 10 trees were cored on the infiltrated plots and five pines outside the experimental area, but within 50 m of it, were cored as control trees. Annual ring widths were measured to an accuracy of 0.01 mm.

2.4. Statistical analyses

All statistical analyses were completed using R version 3.4.2 (R Core Team, 2017). T-tests were used to compare treatment (n = 2) means for different soil layers for total C, N, Ca, K, Mg and Al concentrations, pH (H_2O) and N processes and pools. Results for the N mineralization, nitrification, NH_4-N and NO_3-N pools were log-transformed with a constant of 1 ($\log_{x+1}$) prior to the t-tests to meet the requirement of normal distribution and equal variances.

All plant cover data was arcsine-transformed prior to statistical analysis. T-tests were used for comparing the cover of different plant functional groups (n = 4) and treatments (n = 2) sampled in 2015. Two-way ANOVA was used for comparing the two treatments and the sampling years (1998, 2000, 2001, 2002, 2003) for different plant functional groups and tree ring growth. Pairwise comparisons (using Tukey HSD) were used to find the difference between the treatments and sampling years.

Simpson's diversity index (D) was calculated for each treatment with *diversity* function from the package *vegan* in R (Oksanen, 2017; R Core Team, 2017):

$$D = 1 - \sum p_n^2 \tag{1}$$

where p is the cover of species n in proportion to the cover of all species in the experimental plot.

3. Results

3.1. Soil pH, acidity and nutrients

Soil pH was significantly higher at the previously infiltrated plots in all layers except in the deepest sample from the mineral soil, at 20–40 cm depth (Fig. 1A). Exchangeable acidity, i.e. the amount of acid cations Al^{3+} and H^+ in the soil (and not in the soil solution), was significantly lower at the infiltrated plots in the organic layer (humus) and first 10 cm of the mineral soil (Fig. 1B).

Concentrations of exchangeable Ca^{2+}, K^+, Mg^{2+} and Al^{3+} (mg kg^{-1} d.w.) were different between the treatments, particularly in the deeper layers of the soil (Table 1). Ca^{2+} and Mg^{2+} concentrations were significantly higher post-infiltration than at the control plots in 0–5 and 10–40 cm of the mineral soil. In contrast, exchangeable Al^{3+} concentrations were much higher at the control plots (Table 1).

3.2. N Transformations

Infiltration did not have a significant effect on C and total N (N_{tot}) concentrations (Table 1). In contrast to N_{tot}, significant changes in NH_4-N and NO_3-N concentrations were detected at the infiltrated plots. The rate of net N mineralization (per SOM) was considerably higher at the infiltrated plots both in the organic layer and in the mineral soil ($p = 0.002$, Fig. 2C). Also, NH_4-N concentration (per SOM) was higher in the organic layer of the infiltrated plots ($p = 0.037$, Fig. 2A). Some net nitrification occurred in the mineral soil at the infiltration plots, but the rate was very low as was also the NO_3-N concentration of the soil (Fig. 2B, 2D).

No statistically significant differences were observed in the C:N-ratio of the soil nor in the organic matter (OM) content of the soil organic layer (data not shown). OM % in the soil organic layer tended to be higher at the control plots than at the infiltrated plots, 59% and 46%, respectively. C:N-ratio of the organic layer was 26 at the infiltrated plots and 29 at the control plots.

3.3. Understory vegetation and change in plant cover

In the plant survey completed in 2015, the cover of grasses and forbs tended to be higher at the infiltrated plots ($p = 0.076$), and in contrast, the cover of shrubs and tree seedlings tended to be lower ($p = 0.052$) (Fig. 3). The total number of species tallied at the control and infiltrated plots were 14 and 22, respectively. Simpsons diversity index was higher in the control plots than at the infiltrated plots (Table 2). Finally, the control plots had more litter (including woody and leaf litter) than the infiltrated plots at the time of 2015 sampling, 35.4% and 29.5% respectively. Most abundant plant species, irrespective of the treatment, were *Pleurozium schreberi, Vaccinium myrtillus* and *Vaccinium vitis-idaea* (Table 2).

The different plant survey years and the cover of each plant functional groups were also compared in terms of the effect of infiltration. For mosses, there was a significant treatment and year interaction, and pairwise comparisons revealed a difference in abundance in 2001, 2002 and 2003 ($p = 0.005$, $p = 0.002$, $p = 0.002$, respectively, Fig. 4A). Similarly, for grasses, there was a significant difference between treatments in 2002 ($p = 0.006$, Fig. 4D). Small shrubs and tree seedlings, and lichens showed an overall treatment effect on the abundance of these species but pairwise comparisons between the years did not reveal a difference (Fig. 4B and C).

3.4. Tree growth

Tree growth measurements from the infiltrated plots were compared with the results of control trees outside of the experimental plots. Prior to the infiltration treatment, the pattern of radial growth was similar between the sample trees from the infiltrated plots and the control trees (Fig. 5B). Tree growth indicated a strong response to the added lake water as radial growth of the study trees peaked during and immediately after infiltration, thus infiltration had a significant effect on tree ring growth (Fig. 5A). During and after the infiltration in 1999–2001, tree ring growth roughly doubled on the infiltrated plots, and was still higher at the time of this study (2012–2013). Furthermore, the growth difference between the control trees and treated trees was statistically significant ($p \leq 0.05$) for every year from 2000 onwards (Fig. 5A).

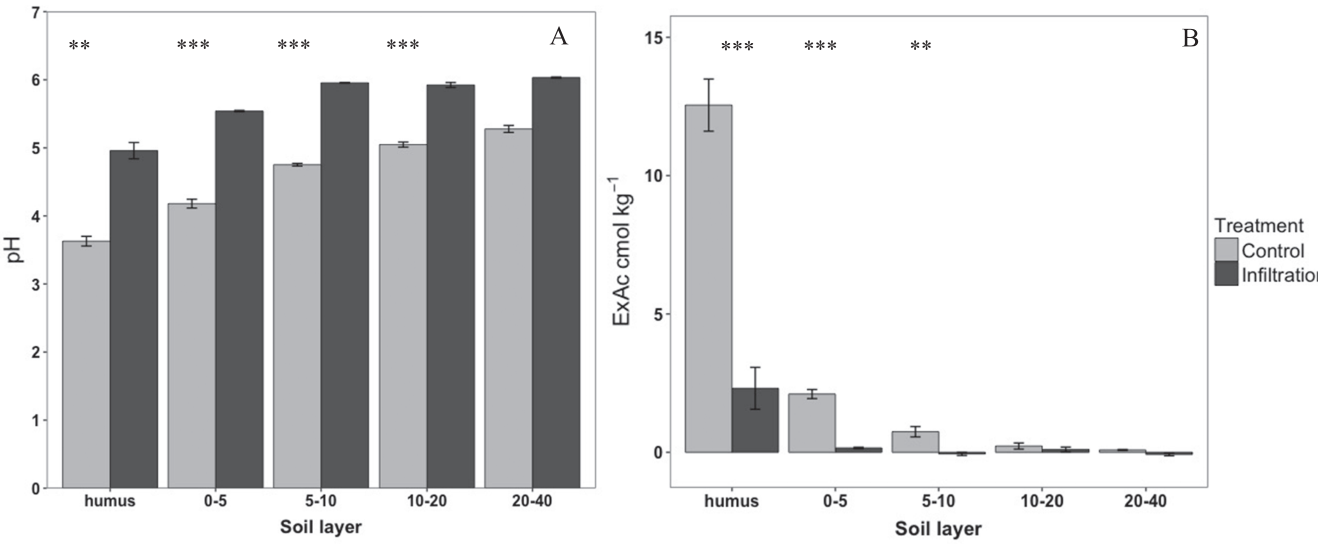

Fig. 1. Soil pH (A) and exchangeable acidity (B) (cmol kg^{-1}) (± SE) for organic layer (humus) and mineral soil layers (cm) for the two different treatments in 2013, 12 years after the infiltration had stopped. Infiltration plots had a significantly higher pH compared to the control plots in all soil layers, except in the deepest mineral soil layer ($p = 0.009$, $p = 0.002$, $p = 0.000$, $p = 0.000$, $p = 0.72$, respectively). Exchangeable acidity was significantly higher at the control plots in the humus layer and the first 10 cm of the mineral soil ($p = 0.000$, $p = 0.000$, $p = 0.019$, respectively). Asterisks indicate statistical significance for the difference of the treatment means ($p \leq 0.05$*, $p \leq 0.01$**, $p \leq 0.001$***).

Table 1
Concentrations of C (g kg^{-1}) and total N (mg kg^{-1}) and exchangeable Ca^{2+}, K^{+}, Mg^{2+} and Al^{3+} (mg kg^{-1}) (± SE) for the organic layer (humus) and mineral soils depths 0–5, 5–10, 10–20, 20–40 cm in 2013, 12 years after the infiltration was stopped.

		Control		Infiltration		p-value*
C g kg^{-1}	humus	230	(18)	170	(45)	0.26
	0–5 cm	17.7	(0.77)	15.9	(2.5)	0.498
	5–10 cm	11.6	(1.25)	11.3	(0.98)	0.84
	10–20 cm	4.6	(1.35)	6.9	(0.51)	0.18
	20–40 cm	2.0	(0.32)	3.6	(0.72)	0.105
N mg kg^{-1}	humus	8153	(890)	6352	(1444)	0.329
	0–5 cm	721	(37)	630	(88)	0.397
	5–10 cm	367	(46)	408	(80)	0.676
	10–20 cm	171	(81)	343	(41)	0.123
	20–40 cm	78	(42)	120	(54)	0.55
Ca mg kg^{-1}	humus	943	(187)	3528	(660)	0.25
	0–5 cm	13.8	(1.30)	313	(56)	0.02*
	5–10 cm	1.7	(0.70)	221	(92)	0.97
	10–20 cm	1.9	(0.80)	83.5	(12)	0.006**
	20–40 cm	1.6	(7.5)	50.9	(7.5)	0.006**
K mg kg^{-1}	humus	266	(37)	318	(65)	0.507
	0–5 cm	11.7	(1.61)	17.7	(4.29)	0.24
	5–10 cm	3.8	(1.01)	10.4	(2.84)	0.71
	10–20 cm	2.1	(0.65)	4.0	(1.15)	0.192
	20–40 cm	1.0	(0.12)	3.5	(0.53)	0.001***
Mg mg kg^{-1}	humus	111	(20.7)	388	(133)	0.128
	0–5 cm	2.8	(0.31)	34.4	(9.44)	0.044*
	5–10 cm	0.5	(0.24)	26.2	(10.62)	0.093
	10–20 cm	0.4	(0.13)	12.5	(2.05)	0.001***
	20–40 cm	0.3	(1.34)	8.3	(2.04)	0.041*
Al mg kg^{-1}	humus	180	(40)	38	(16)	0.017*
	0–5 cm	74.5	(5.72)	6.2	(1.65)	0.001***
	5–10 cm	26.8	(8.81)	6.9	(3.87)	0.084
	10–20 cm	11.0	(3.82)	2.7	(0.33)	0.074
	20–40 cm	4.7	(1.33)	3.1	(1.10)	0.887

All values are per dry weight (d.w.).
Statistical significance for the difference of the treatment means ($p \leq 0.05$, $p \leq 0.01$**, $p \leq 0.001$***).

4. Discussion

In this study, we found that pH had remained high for 13 years after the infiltration treatment had been stopped, particularly in the mineral soil, having reduced from 6.7 to 5.4 in the organic layer and from 6.3 to 5.7 in the mineral soil (Fig. 1A). During the infiltration in 1999–2001, soil pH rose significantly at the infiltrated plots. Nöjd et al. (2009) reported an increase from pH 3.8 to 6.7 from 1998 to 2001. Similar results have been reported at other sprinkling infiltration sites (Paavolainen et al., 2000a; Lindroos et al., 2001).

In 2013, the concentrations of base cations, Ca^{2+} and Mg^{2+} were still significantly higher in the infiltrated soil, particularly in the deeper soil layers (Table 1). In contrast, the effect of infiltration on K^{+} was less pronounce. As a relatively mobile nutrient with low affinity to cation exchange sites, K^{+} easily leaches with the percolating water from the soil. These results are in agreement with the findings of Lindroos et al. (2001) and Nöjd et al. (2009) who found that the saturation of the cation exchange sites by base cations occurred soon after the start of the infiltration. Furthermore, Nöjd et al. (2009) found no decline in Ca^{+2} and Mg^{2+} two years after the termination of infiltration. It appears that the hydrogen ions and Al^{3+} were relatively quickly displaced from the soil cation exchange sites by Ca^{2+} and Mg^{2+}. At the control plots, both the concentration of Al^{3+} and exchangeable acidity were higher, thus indicating that the exchange sites were controlled more by Al^{3+} than by base cations. The amount of free Al^{3+}, and Al^{3+} bound at the exchange sites or to the organic matter complexes is closely related to the soil pH.

After the 13-year recovery period, soil net N mineralization and soil NH_4-N concentration were significantly higher on the infiltrated plots than on the control plots. In the organic layer, net N mineralization was negative at the start of the infiltration and had remained negative to the end of the infiltration. However, infiltration increased the NH_4-N concentration. Nöjd et al. (2009) concluded that the main driver for this increase was the NH_4-N added to the soil in the sprinkled lake water. In the mineral soil, NO_3-N concentrations had increased significantly at the end of the infiltration in 2001. Previous studies on more fertile sites

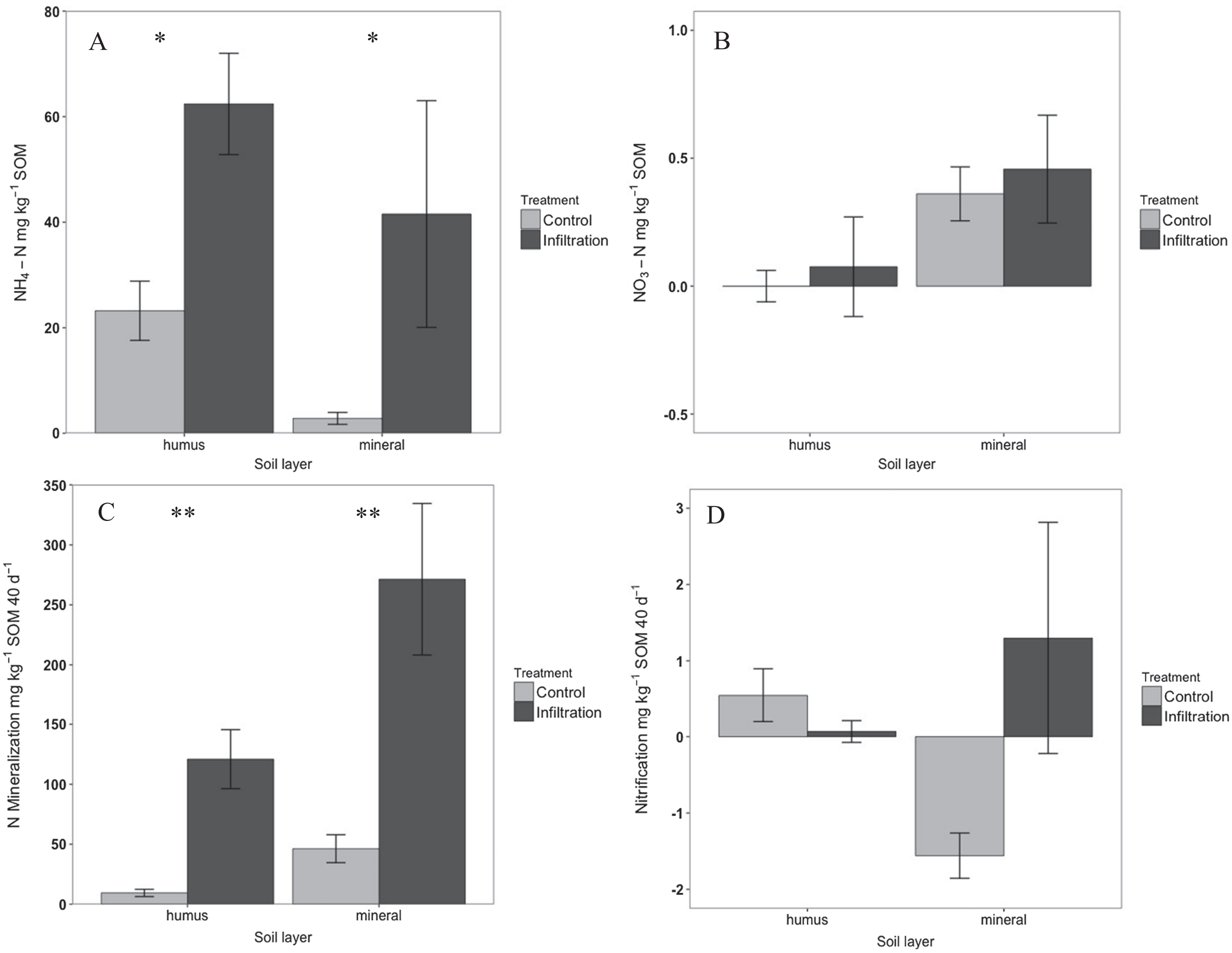

Fig. 2. NH_4-N (A) and NO_3-N (B) (± SE) concentrations and rates of net N mineralization (C) and net nitrification mg kg^{-1} (D) (± SE) for the organic layer (humus) and mineral soil (0–10 cm) in 2014, 13 years after the infiltration had stopped. All values are per soil organic matter (SOM). The rate of net N mineralization and NH_4-N were significantly higher at the infiltrated plots. Asterisks indicate statistical significance for the difference of the treatment means ($p \le 0.05$*, $p \le 0.01$**, $p \le 0.001$***).

have shown that infiltration initiates nitrification (Paavolainen et al., 2000a, 2000b; Nöjd et al., 2009). Interestingly, this effect had previously only been observed in the mineral soil (Paavolainen et al., 2000a). This increase in nitrification was probably very much linked to the higher soil pH. Though the net N mineralization rate was high in 2014 in previously infiltrated soil, the rate of net nitrification was low (Fig. 2). Paavolainen et al. (2000a, 2000b) reported that when excess of NH_4-N was present in the soil, soil pH was the major factor in determining the rate of NO_3-N production, and concluded that the nitrifiers at the infiltrated plots were more sensitive to changes in pH and adapted to a higher pH than their counterparts at the control plots. The study by Paavolainen et al. (2000b) was carried out in a highly fertile site in southern Finland, where nitrification had remained high two-years after the termination of the infiltration.

At Vuontee, the median N concentration (NH_4-N 0.002 mg/liter, NO_3-N 0.07 mg/l and N_{tot} 0.38 mg/l) of the added lake water was relatively low, however due to the large quantities added, the annual C, N and nutrient accumulation in the soil can be significant during treatment (Helmisaari et al., 2003). At an experimental esker in Southern Finland (Lindroos et al., 2001), also infiltrated with lake water in the 1990s, an estimated 10 Mg of C was added to the vadose zone in one year (Helmisaari et al., 2003). The N concentrations of the percolate water and groundwater were measured during the infiltration in 1999–2001 (Helmisaari et al., 1999). Throughout the treatment and after its termination NO_3-N and DON (dissolved organic nitrogen) levels in the groundwater remained below the EU-limit (0.5 mg/l) (Helmisaari et al., 1999; European Commission, 2010).

Nitrogen addition, soil pH and NH_4-N availability are amongst the environmental factors which control and alter the composition, abundance and diversity of the nitrifying microorganism community (ammonia-oxidizing bacteria and archaea) in the soil (Levy-Booth et al., 2014). Contrary to intensive nitrification on more fertile sites due to infiltration (Paavolainen et al., 2000a), nitrification did not dominate at this site in such a high rate (Fig. 2). So why was net nitrification at the infiltrated plots low even though the conditions in the soil seemed to be ideal at the time of this study in 2014? Low nitrate concentration could be explained by uptake of nitrate by the understory vegetation, but this does not explain the low net nitrification. One possible explanation for the net nitrification occurring predominantly in the mineral soil, could be the lack of inhibitory organic compounds, such as terpenes, occurring in high concentrations in the organic layer (Smolander et al., 2012).

Simpson's diversity indices for the two different treatments were different. Simpson's diversity index takes into account both species richness and evenness of the abundance of the species. The control plots had higher diversity indices, however a species tally did reveal that the

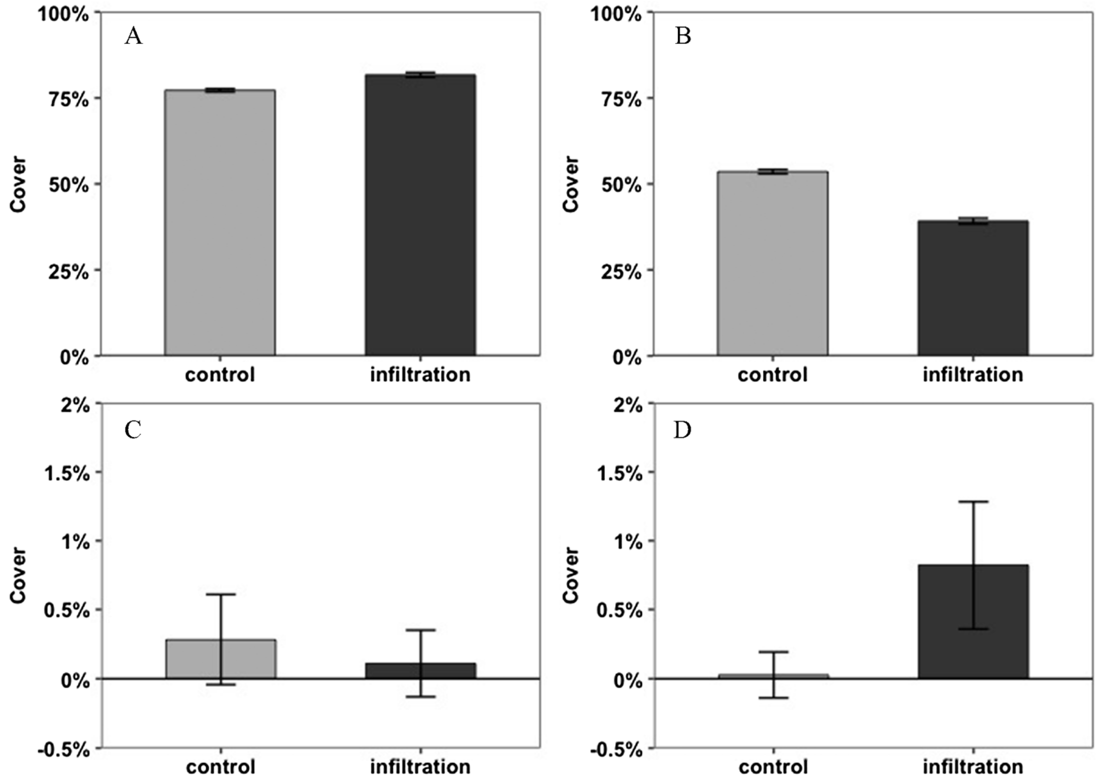

Fig. 3. Cover of the plant functional groups (%, ± SE): mosses (A), dwarf shrubs and tree seedlings (B), lichens (C) and forbs and grasses (D), 14 years after the termination on infiltration in 2015 at the control plots (light grey) and the infiltrated plot (dark grey).

Table 2
Simpson's diversity index, plant functional groups and mean plant species cover (%) per treatment in 2015, 14 years after the infiltration had stopped. The list represents all the species identified in 2015 vegetation survey.

Plant functional group	Species	Control	Infiltration
Simpson's index	–	0.69	0.62
Dwarf shrubs and tree seedlings	*Betula pendula*	0	0.06
	Picea abies	0	0.02
	Salix sp.	0	0.06
	Diphasiastrum complanatum	0	0.02
	Pinus sylvestris	0.22	0.38
	Empetrum nigrum	4.39	0.68
	Vaccinium myrtillus	37.7	29.3
	Vaccinium vitis-idaea	11.5	8.2
	Calluna vulgaris	0.28	0.33
Grasses and forbs	*Melampyrum pratense*	0.03	0.80
	Melampyrum sylvaticum	0	0.02
	Orthilia secunda	0	0.06
Mosses	*Tilium crista-castrensis*	5.08	0
	Polytrichum commune	0	0.03
	Pleurozium schreberi	59.1	68.4
	Hylocomium splendens	10.2	6.8
	Dicranum polysetum	0.95	3.39
	Dicranum fuscescens	1.87	3.02
	Brachythecium spp.	0	0.01
Lichens	*Cladina stellaris*	0.21	0.02
	Cladina rangiferina stygia	0.05	0.03
	Cetraria islandica	0.03	0.06

infiltration plots had higher individual plant species richness (i.e. number of species), thus it was likely that the lower species evenness was reflected to the Simpson's index. Some species were only present at the infiltrated plots, such as *Betula pendula*, *Polytrichum commune* and *Salix spp*, whereas species such as *Vaccinium myrtillus* were abundant at all plots irrespective of whether they were infiltrated or not.

Based on the findings of this study, it seems that sprinkling infiltration favored early-successional forbs and grasses (*Epilobium* spp.) and negatively affected late successional, slow-growing mosses and lichens. Previous studies from the infiltrated sites reported a reduction in the cover of lichens and mosses during 1998–2003, and an emergence of new herbs and grasses, such as *Epilobium* spp. and *Tussilago farfara*, and finally, a gradual increase in the cover of dwarf shrubs at all plots, irrespective of treatment (Derome et al., 2004). In the survey carried out in 2015, we found that the moss cover had recovered from the infiltration. Forbs and herbs such as grasses too were still more abundant at the infiltrated plots. In contrast, the cover of lichens had reduced at all plots. The added moisture and standing water, together with competition by herbs, likely contributed to the decrease in the cover of lichens at the infiltrated plots.

The changes in the vegetation composition caused by sprinkling infiltration are probably a result of both the altered moisture conditions and N added in the infiltration water. Sprinkling infiltration entails a certain dichotomy: as a treatment it is both a form of fertilizing and due to the large quantities of added water, also a disturbance. Several studies have assessed the effects of N addition on understory vegetation, however only a few have studied the effects of added nutrient-rich water. Long-term monitoring experiment completed in 28 European forest sites found that the changes in species composition resulting from an increased atmospheric N deposition suggest that oligotrophic species decreased in cover, and were thus more sensitive to added N (Dirnböck et al., 2014). Nitrogen fertilization and disturbance seem to have similar effects on understory vegetation, however different species can be more susceptible to either-or, or both (Saarsalmi and Mälkönen, 2001; Manninen et al., 2009; Metcalfe et al., 2013). Manninen et al. (2009) reported dwarf shrubs such as *Vaccinium myrtillus* and *Vaccinium vitis-idea* being more susceptible to physical disturbance than to N increase (fertilization or deposition). However, the findings of Metcalfe et al. (2013) led them to conclude, that whether added N increases the abundance of specific species, is still debatable. In their study, *V.*

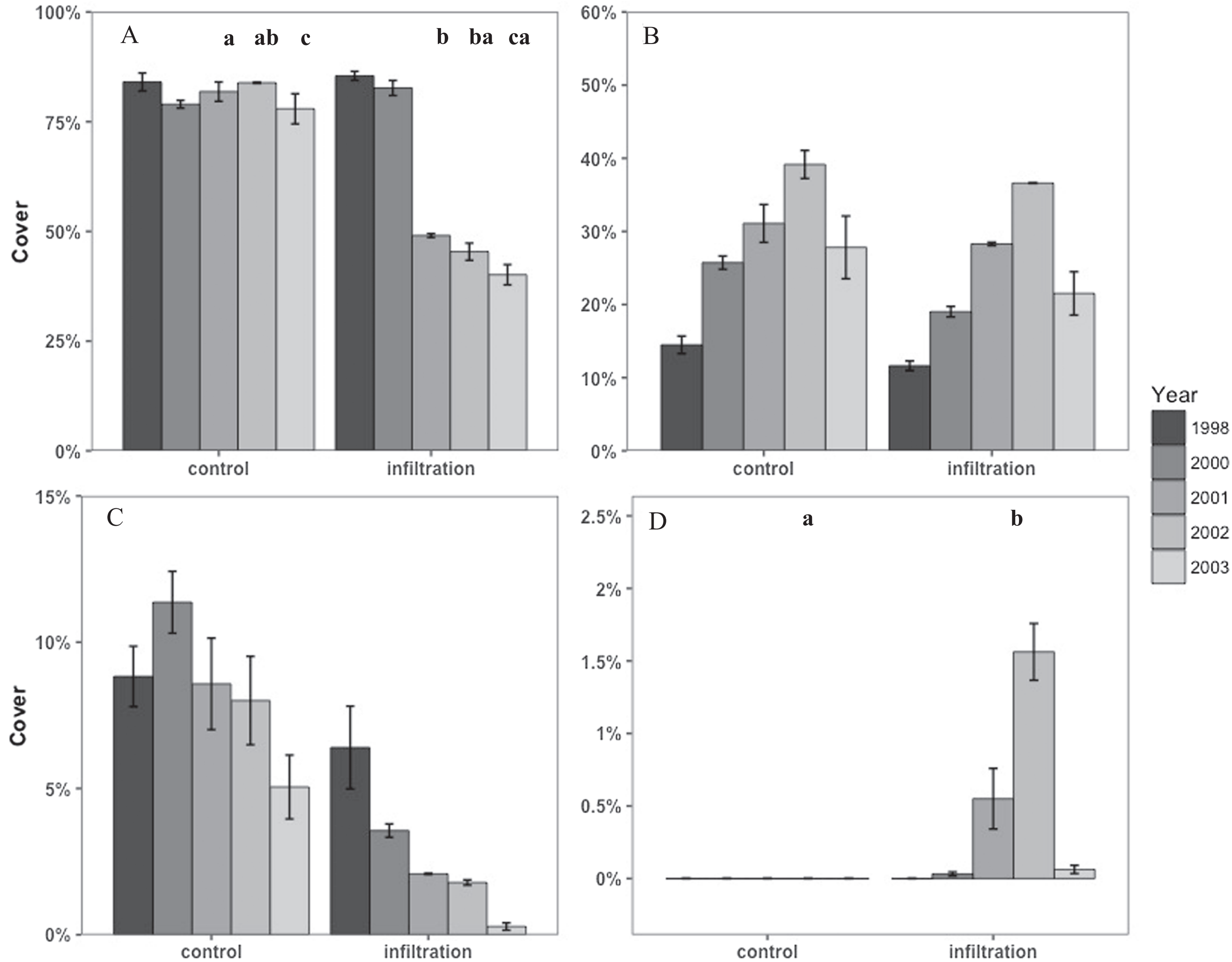

Fig. 4. Cover of the plant functional groups (%): mosses (A), dwarf shrubs (B), lichens (C) and herbs and grasses (D) (± SE) for different years; before (1998), during (2000–2001) and after (2002–2003) the infiltration. Different letters indicate a significant difference between the treatments and years ($p \leq 0.05$).

myrtillus responded positively to added N (fertilization), which was in contrast to the other studies that had observed a reduction in *V. myrtillus* cover following long-term N additions (Metcalfe et al., 2013). In Finland, forest N fertilization has been reported to increase the cover of grasses and some early-successional herbs such as *Rubus idaeus* (Saarsalmi and Mälkönen, 2001). Mosses, in contrast, have been reported to respond negatively to N fertilization (Saarsalmi and Mälkönen, 2001) and large scale disturbance, such as logging (Tarvainen et al., 2015; Tonteri et al., 2016). As a result of logging, inorganic N becomes increasingly more available for the remaining plants in the stand (Likens et al., 1970; Kreutzweiser et al., 2008; Schelker et al., 2016) and nutrient uptake by the ground vegetation following a clear-cut can be significant sink of N and other nutrients (Olsson and Staaf, 1995). In a boreal coniferous stand in Finland, Palviainen et al. (2005) reported a delay of two years following harvest during which the ground vegetation re-emerges and recovers and nutrient leaching losses from the soil are the greatest (Piirainen et al., 2002, 2004).

Irrigation experiments in the boreal region are rare. Bergh et al. (1999) found that irrigation (combined with fertilization) more than doubled the growth of Norway spruce at a fertile, southern Swedish stand fertilized without irrigation. Fertilization effects on tree growth have been extensively studied in the Nordic region, and tree growth increases following a N fertilization have been reported both in Finland (Saarsalmi and Mälkönen, 2001) and Sweden (Bergh et al., 2014; Sponseller et al., 2016). It is also important to note that the duration of commercial N fertilization (i.e. growth response to added N) depends very much on the dosage and on the forest site-type, and can vary between 7–10 years, as reported from Finnish Scots pine stands (Kukkola and Saramäki, 1983; Saarsalmi and Mälkönen, 2001). Furthermore, past studies have indicated that in the forest soil, N fertilization produces different responses in terms of N and C mineralization; long-term fertilization can reduce C mineralization and increase that of N (Olsson et al., 2005; Saarsalmi et al., 2012). In a study by Kukkola and Saramäki (1983) the largest growth increase was observed 2–3 years after the fertilization treatment, after which tree growth gradually decreased. The growth increase observed at Vuontee follows a somewhat similar pattern (Fig. 5); radial growth of the study trees peaked during and immediately after infiltration. Nöjd et al. (2009) accounted the observed increase in tree growth to the increased availability of N and water in the soil. They concluded, however, that pinpointing which of these two was more responsible for the growth response is difficult (Nöjd et al., 2009). Typically, these site types are both water- and nutrient-limited at the time of maximum growth, and water availability fluctuates according to season. This initial intense “fertilizing effect” of the infiltration lasted about five years after the initiation of the treatment. However, tree ring growth was still higher on the infiltrated plots in 2013 (Fig. 5). It should be mentioned that Nöjd et al. (2009) reported an outbreak of European pine sawfly (*Neodiption sertifer*) in 1981, which resulted in a sharp decrease in tree growth in the experimental stand.

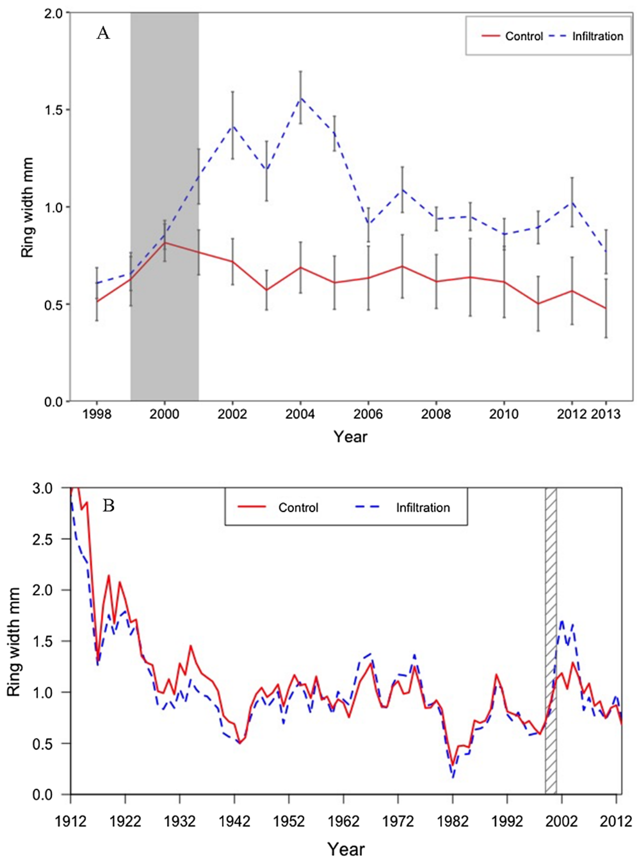

Fig. 5. Mean annual radial growth (mm, ± SE) of Scots pines in the infiltrated area (blue dotted line, n = 10 trees) and in the control plots (red solid line, n = 5 trees). The time period shown is 1998–2013 (A) and 1912–2013 (B). The highlighted grey area indicates the period of infiltration (1999–2001). Tree ring growth was significantly higher ($p \leq 0.05$) in the infiltrated trees compared to the controls trees every year from 2000 to 2013.

Evidently, sprinkling infiltration caused chemical changes in the forest soil and contributed to the changes in the understory vegetation and finally affected tree growth in the stand. All these observations should be taken into consideration when planning new infiltration plants – areas that should preferably remain in their original state should be avoided. Finally, scientific experiments tend to utilize carefully controlled and monitored experimental set-ups. This study was planned in cooperation with the local waterworks who still operate the infiltration water plant. Research on an operating infiltration plant increased the practicality of the study, but caused a few unexpected problems in the field sampling, as was discussed earlier.

From a long-term perspective, it remains to be established how infiltrated forest stands are able to persist with the C and N additions brought upon by the watering treatment. The purpose of groundwater recharge plants – this applies to all household water production – is to produce high-quality drinking water. The forest areas reserved for producing household water are managed mainly for that purpose, thus they are water production areas and no longer considered natural forests or productive stands managed solely for wood production or recreational purposes. These facts restrict what areas are suitable for sprinkling infiltration in the future.

5. Conclusions

Sprinkling infiltration adds large amounts of nutrients and organic matter to the forest floor and soil. Soil recovery can be a relatively slow process, particularly on sites where the soil cation exchange capacity has largely shifted from its original state. Acidity is added to the soil through only precipitation and weathering. In this study, the effects of sprinkling infiltration had persisted at the experimental site that had previously been infiltrated with lake water; soil pH and the concentrations of base cations Ca^{2+} and Mg^{2+} had all remained high at the infiltrated plots 12 years after the termination of the treatment. The rate of net N mineralization was much higher at the infiltrated plots both in the organic layer and in the mineral soil, whereas nitrification was negligible. These results lead to the conclusion that sprinkling infiltration results in the long-term neutralization of the forest soil. The large amounts of added water created conditions unfavorable to certain plant species such as lichens, but favored early-successional herbs, grasses and forbs, and finally – trees.

Sprinkling infiltration as a groundwater recharge method is an environment altering soil treatment which, based on the findings of this study, can have long-term effects on tree growth, soil processes and understory vegetation. Infiltration induced changes in soil dynamics could potentially be long-lasting and soil recovery from such a treatment may take years, if not decades.

Acknowledgements

This study was funded by Maj and Tor Nessling Foundation, Niemi-foundation and Maa - ja vesitekniikan tuki ry, and the Finnish Society of Forest Science**.** We are grateful to the Waterworks of Jyväskylä (at present Jyväskylä Energy Group) for allowing us to carry out our experiment at Vuontee-esker. Also, we wish to thank Veikko Ritvaniemi, the owner of the forest area of our research plots, for his interest and valuable observations at the site.

References

Bergh, J., Linder, S., Lundmark, T., Elfving, B., 1999. The effect of water and nutrient availability on the productivity of Norway spruce in northern and southern Sweden. For. Ecol. Manage. 119, 51–62. https://doi.org/10.1016/S0378-1127(98)00509-X.
Bergh, J., Nilsson, U., Allen, H.L., Johansson, U., Fahlvik, N., 2014. Long-term responses of Scots pine and Norway spruce stands in Sweden to repeated fertilization and thinning. For. Ecol. Manage. 320, 118–128. https://doi.org/10.1016/j.foreco.2014.02.016.
Cajander, A.K., 1949. Forest types and their significance. Acta For Fenn. 56, 1–71.
Derome, Helmisaari, H.-S., Korpela, L., Lindroos, A.-J., Lumme, I., Nieminen, T.M., Nöjd, P., 2004. Sadetusimeytyksen vaikutukset Vuonteenharjulla - maaperän ja kasvillisuuden toipuminen imeytyksestä. METLA, Vantaan tutkimuskeskus, Vantaa (In Finnish).
Derome, J., Lindroos, A.-J., Helmisaari, H.-S., 2006. Effect of sprinkling infiltration on soil acidity and fertility properties on a forested esker in Central Finland. UNESCO IHO-VI Series Groundwater 13, 264–268.
Dirnböck, T., Grandin, U., Bernhardt-Römermann, M., Beudert, B., Canullo, R., Forsius, M., Grabner, M.T., Holmberg, M., Kleemola, S., Lundin, L., Mirtl, M., Neumann, M., Pompei, E., Salemaa, M., Starlinger, F., Staszewski, T., Uzieblo, A.K., 2014. Forest floor vegetation response to nitrogen deposition in Europe. Glob. Chang. Biol. 20, 429–440. https://doi.org/10.1111/gcb.12440.
European Commission, 2010. The EU Nitrates Directive. European Environment Agency. (Available at: ec.europa.eu-environment-pubs-pdf-factsheets-nitrates.pdf).
GTK, 2016. Geological Survey of Finland. Available at: < http://en.gtk.fi/land_use/groundwater.html > (Accessed 06/2017).
Helmisaari, H.-S., Derome, J., Kitunen, V., Lindroos, A.-J., Lumme, I., Monni, S., Nöjd, P., Paavolainen, L., Pesonen, E., Salemaa, M., Smolander, A., 1999. Veden imeytyksen vaikutukset metsämaahan ja kasvillisuuteen sekä vajo- ja pohjaveden laatuun. Metsäntutkimuslaitoksen tiedonantoja 721 [Finnish Forest Research Institute, Research Papers]. 721, 72, 96. (In Finnish).
Helmisaari, H.-S., Illmer, K., Hatva, T., Lindroos, A.-J., Miettinen, I., Pääkkönen, J., Reijonen, R. (Eds.), 2003. Tekopohjaveden muodostaminen: imeytystekniikka, maaperäprosessit ja veden laatu, TEMU-tutkimushankkeen loppuraportti. Metsäntutkimuslaitoksen tiedonantoja 902 [Finnish Forest Research Institute, Research Papers], Vantaa, Finland. (In Finnish).
Helmisaari, H.-S., Illmer, K., Derome, J., Hatva, T., Kitunen, V., Lindroos, A.-J., Miettinen, I., Pääkkönen, J., Reijonen, R., 2006. Artificial recharge in Finland through basin and sprinkling infiltration: soil processes, retention time and water quality. UNESCO IHO-VI Series Groundwater 13, 280–285.
Hämet-Ahti, L., Suominen, J., Ulvinen, T., Uotila, P. (Eds.), 1998. Retkeilykasvio (Field flora of Finland), fourth ed. Finnish Museum of Natural History, Botanical Museum. Helsinki (in Finnish).
Högberg, P., Näsholm, T., Franklin, O., Högberg, M.N., 2017. Tamm Review: on the nature of the nitrogen limitation to plant growth in Fennoscandian boreal forests. For. Ecol. Manage. 403, 161–185. https://doi.org/10.1016/j.foreco.2017.04.045.
Korhonen, J.F.J., Pihlatie, M., Pumpanen, J., Aaltonen, H., Hari, P., Levula, J., Kieloaho,

A.-J., Nikinmaa, E., Vesala, T., Ilvesniemi, H., 2013. Nitrogen balance of a boreal Scots pine forest. Biogeosciences 10, 1083–1095.

Kreutzweiser, D.P., Hazlett, P.W., Gunn, J.M., 2008. Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: a review. Environ Rev. 16, 157–179. https://doi.org/10.1139/A08-006.

Kukkola, M., Saramäki, J., 1983. Growth response in repeatedly fertilized pine and spruce stands on mineral soils. Communicationes Instituti Forestalis Fenniae 114, 55.

Kummu, M., Guillaume, J.H.A., de Moel, H., Eisner, S., Flörke, M., Porkka, M., Siebert, S., Veldkamp, T.I.E., Ward, P.J., 2016. The world's road to water scarcity: shortage and stress in the 20th century and pathways towards sustainability. Sci Rep. 6, 38495. https://doi.org/10.1038/srep38495.

Kätkö, T.S., Lipponen, M.A., Rönkä, E.K.T., 2006. Groundwater use and policy in community water supply in Finland. Hydrogeol. J. 14, 69–78. https://doi.org/10.1007/s10040-004-0351-3.

Levy-Booth, D.J., Prescott, C.E., Grayston, S.J., 2014. Microbial functional genes involved in nitrogen fixation, nitrification and denitrification in forest ecosystems. Soil Biol. Biochem. 75, 11–25. https://doi.org/10.1016/j.soilbio.2014.03.021.

Likens, G.E., Bormann, F.H., Johnson, N.M., Fisher, D.W., Pierce, R.S., 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook Watershed-Ecosystem. Ecol. Monog. 40, 23–47.

Lindroos, A.-J., Derome, J., Paavolainen, L., Helmisaari, H.-S., 2001. The effect of lake water infiltration on the acidity and base cation status of forest soil. Water Air Soil Poll. 131, 153–167.

Lindroos, A.-J., Kitunen, V., Derome, J., Helmisaari, H.-S., 2002. Changes in dissolved organic carbon during artificial recharge of groundwater in a forested esker in Southern Finland. Water Res. 36, 4951–4958. https://doi.org/10.1016/S0043-1354(02)00226-9.

Manninen, O.H., Stark, S., Kytöviita, M.M., Lampinen, L., Tolvanen, A., 2009. Understorey plant and soil responses to disturbance and increased nitrogen in boreal forests. J. Veg. Sci. 20, 311–322. https://doi.org/10.1111/j.1654-1103.2009.05529.x.

Metcalfe, D.B., Eisele, B., Hasselquist, N.J., 2013. Effects of nitrogen fertilization on the forest floor carbon balance over the growing season in a boreal pine forest. Biogeosciences. 10, 8223–8231.

Nöjd, P., Lindroos, A.-J., Smolander, A., Derome, J., Lumme, I., Helmisaari, H.-S., 2009. Artificial recharge of groundwater through sprinkling infiltration: Impacts on forest soil and the nutrient status and growth of Scots pine. Sci. Total Environ. 407, 3365–3371. https://doi.org/10.1016/j.scitotenv.2009.01.062.

Oksanen, J., 2017. Multivariate analysis of ecological communities in R: Vegan tutorial. R package version 2.4.4.

Olsson, B., Staaf, H., 1995. Influence of harvesting intensity of logging residues on ground vegetation in coniferous forests. J Appl Ecol. 32, 640–654.

Olsson, P., Linder, S., Giesler, R., Högberg, P., 2005. Fertilization of boreal forest reduces both autotrophic and heterotrophic soil respiration. Glob. Chang. Biol. 11, 1745–1753. https://doi.org/10.1111/j.1365-2486.2005.001033.x.

Paavolainen, L., Smolander, A., 1998. Nitrification and denitrification in soil from a clear-cut norway spruce (Picea abies) stand. Soil Biol. Biochem. 30, 775–781.

Paavolainen, L., 1999. Nitrogen transformations in boreal forest soils in response to extreme manipulation treatments. Finnish Forest Research Institute, Research Papers 746, Vantaa, Finland.

Paavolainen, L., Fox, M., Smolander, A., 2000a. Nitrification and denitrification in forest soil subjected to sprinkling infiltration. Soil Biol. Biochem. 32, 669–678.

Paavolainen, L., Smolander, A., Lindroos, A.-J., Derome, J., Helmisaari, H.-S., 2000b. Nitrogen transformations and losses in forest soil subjected to sprinkling infiltration. JEQ 29, 1069–1074.

Palviainen, M., Finér, L., Mannerkoski, H., Piirainen, S., Starr, M., 2005. Changes in the above- and below-ground biomass and nutrient pools of ground vegetation after clear-cutting of a mixed boreal forest. Plant Soil. 275, 157–167. https://doi.org/10.1007/s11104-005-1256-1.

Piirainen, S., Finér, L., Mannerkoski, H., Starr, M., 2002. Effects of forest clear-cutting on the carbon and nitrogen fluxes through podzolic soil horizons. Plant Soil. 239, 301–311.

Piirainen, S., Finér, L., Mannerkoski, H., Starr, M., 2004. Effects of forest clear-cutting on the sulphur, phosphorus and base cation fluxes through podzolic soil horizons. Biogeochemistry. 69, 405–424.

Priha, O., Smolander, A., 1997. Microbial biomass and activity in soil and litter under Pinus sylvestris, Picea abies and Betula pendula at originally similar field afforestation sites. Biol. Fertil. Soils. 24, 45–51. https://doi.org/10.1007/BF01420219.

R Core Team, 2017. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Saarsalmi, A., Mälkönen, E., 2001. Forest fertilization research in Finland: a literature review. Scand. J. For. Res. 16, 514–535. https://doi.org/10.1080/02827580152699358.

Saarsalmi, A., Smolander, A., Kukkola, M., Moilanen, M., Saramaki, J., 2012. 30-Year effects of wood ash and nitrogen fertilization on soil chemical properties, soil microbial processes and stand growth in a Scots pine stand. For. Ecol. Manage. 278, 63–70. https://doi.org/10.1016/j.foreco.2012.05.006.

Schelker, J., Sponseller, R., Ring, E., Högbom, L., Löfgren, S., Laudon, H., 2016. Nitrogen export from a boreal stream network following forest harvesting: seasonal nitrate removal and conservative export of organic forms. Biogeosciences 13, 1–12. https://doi.org/10.5194/bg-13-1-2016.

Smolander, A., Kanerva, S., Adamczyk, B., Kitunen, V., 2012. Nitrogen transformations in boreal forest soils-does composition of plant secondary compounds give any explanations? Plant Soil. 350, 1–26. https://doi.org/10.1007/s11104-011-0895-7.

Sponseller, R.A., Gundale, M.J., Futter, M., Ring, E., Nordin, A., Näsholm, T., Laudon, H., 2016. Nitrogen dynamics in managed boreal forests: recent advances and future research directions. Ambio 45, 175–187. https://doi.org/10.1007/s13280-015-0755-4.

Stenroos, S., Ahti, T., Lohtander, K., Myllys, L., 2011. Suomen jäkäläopas. Kasvimuseo, Luonnontieteellinen keskusmuseo, Helsingin yliopisto. 535 p (in Finnish).

Tarvainen, O., Hekkala, A.-M., Kubin, E., Tamminen, P., Murto, T., Tolvanen, A., 2015. Soil disturbance and early vegetation response to varying intensity of energy wood harvest. For. Ecol. Manage. 348, 153–163.

Tonteri, T., Hotanen, J.P., Kuusipalo, J., 1990. The Finnish forest site type approach - ordination and classification studies of mesic forest sites in southern Finland. Vegetatio 87, 85–98.

Tonteri, T., Salemaa, M., Rautio, P., Hallikainen, V., Korpela, L., Merilä, P., 2016. Forest management regulates temporal change in the cover of boreal plant species. For. Ecol. Manage. 381, 115–124. https://doi.org/10.1016/j.foreco.2016.09.015.

Ulvinen, T., Syrjänen, K., Anttila, S. (Eds.), 2002. Suomen sammalet – levinneisyys, ekologia, uhanalaisuus. Suomen ympäristö 560. Luonto ja luonnonvarat. Suomen ympäristökeskus. 353 p. (In Finnish).